

Improving odour assessment in LCA—the odour footprint

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Abstract

Purpose Odour is an important aspect of systems for human and agricultural waste management and many technologies are developed with the sole purpose of reducing odour. Compared with greenhouse gas assessment and the assessment of toxicity, odour assessment has received little attention in the life cycle assessment (LCA) community. This article aims to redress this.

Methods Firstly, a framework for the assessment of odour impacts in LCA was developed considering the classical LCA framework of emissions, midpoint and endpoint indicators. This suggested that an odour footprint midpoint indicator

was worth striving for. An approach to calculating an areal indicator we call “odour footprint”, which considers the odour detection threshold, the diffusion rate and the kinetics of degradation of odourants, was implemented in MATLAB. We demonstrated the use of the characterisation factors we calculated in a case study based on odour removal technology applied to a pig barn.

Results and discussion We produced a list of 33 linear characterisation factors based on hydrogen sulphide equivalents, analogous to the linear carbon dioxide equivalency factors in use in carbon footprinting, or the dichlorobenzene equivalency factors developed for assessment of toxic impacts in LCA. Like the latter, this odour footprint method does not take local populations and exposure pathway analysis into account—its intent is not to assess regulatory compliance or detailed design. The case study showed that despite the need for materials and energy, large factor reductions in odour footprint and eutrophication potential were achieved at the cost of a smaller factor increase in greenhouse emissions.

Conclusions The odour footprint method is proposed as an improvement on the established midpoint method for odour assessment in LCA. Unlike it, the method presented here considers the persistence of odourants. Over time, we hope to increase the number of characterised odourants, enabling analysts to perform simple site-generic LCA on systems with odourant emissions.

Keywords Footprint · LCIA · Midpoint indicator · Odor · Odour · Swine

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1 Introduction

Odour emissions from industrial facilities, agricultural enterprises and sewage treatment systems are an ongoing concern (e.g. Nicell 2009). Odour plays a role in the selection of

technologies for manure and waste water management. Population pressure urbanises rural areas, bringing more homes closer to existing agricultural or industrial odour sources, while high-density animal husbandry operations continue to proliferate. Consequently, the incidence of odour annoyance is increasing (Powers et al. 2005) and it is increasingly important to be able to compare technologies in terms of odorous emissions. In this context, it is reasonable to expect the importance of assessing odour in a life cycle assessment (LCA) framework to increase. In an LCA that ignored odour impacts, a ventilation technology which reduces odourant emissions would always seem unfavourable in a comparison with one that does not reduce odour due to higher material and energy inputs required for odour abatement. In short, if we use the approach to the consideration of novel life cycle impacts suggested by Cucurachi et al. (2014) as a guide and summarise its first four steps, there is evidence that the odour impact category is important and that there is a need to quantify it in relationship to life cycles.

This need has been identified previously (Heijungs et al. 1992), although methodologies for the assessment of odour in an LCA framework have not had the same level of application nor attention to development as other notable frontiers of the framework such as the assessment of freshwater use (Kounina et al. 2013) and chemical toxicity (Goedkoop et al. 2009). Others such as radio-frequency electromagnetic fields and ecological light pollution are at an even earlier stage of consideration (Cucurachi et al. 2014). In the original “Guide and Backgrounds” to LCA (Heijungs et al. 1992) the use of odour threshold values (OTVs) was proposed as a life cycle impact assessment (LCIA) method for “malodorous air”, characterising odourant emissions based on the critical volumes approach. The OTV is the concentration (kg m^{-3}) at which a chemical is detectable compared with clean air by 50 % of the population. Thus, for the mass m of each odourant i ,

$$\text{malodorous air} = \sum_i \frac{m_{i,\text{air}}}{\text{OTV}_{i,\text{air}}} \quad (1)$$

An emission of a mass of an odourant is thus assessed as the volume of air it would occupy when instantaneously diluted to its OTV. The characterisation factors are simply effect factors (the inverse of the OTVs). Conveniently, this approach allows the linkage of a functional unit with relevant elementary flows (step 5 in Cucurachi et al. (2014)). Heijungs et al. (1992) proposed that these effect factors might be a basis for future development of “smell creation potential” characterisation factors which would include a fate factors. But this never happened and Heijungs’ original approach and the 60 characterisation factors (CFs) he based on an older document (Roos 1989) were used and duplicated in the subsequent Nordic LCA guidelines (although with errors—Nord 1995),

and the updated Backgrounds document (Guinee 2002). The method is suggested (without providing data) in the popular Hitch Hiker’s Guide to LCA (Baumann and Tillman 2004). The summary of the most recent European thinking about life cycle impact assessment (LCIA), the ILCD Handbook (JRC 2010), does not provide any guidance on odour (although it says analysts should be prepared to create new LCIA methods where necessary), so the most recent mainstream guidance on odour uses an idea and data which are 25 years old.

Many other researchers (Feilberg et al. 2011; Hansen et al. 2012; Parker et al. 2012) have made use of odour threshold concepts in odour management studies without incorporating other aspects of LCA. A literature search identified only a few case studies in which odour had actually been assessed in the LCA framework (see Table 1). Various other LCA articles make qualitative references to odour impacts (e.g. Bridle and Skrypski-Mantele 2000; Rabl et al. 2008), and it is possible that additional researchers have examined it in the LCA framework without drawing attention to that fact in the abstract or keywords of their articles.

In this article, an overall framework is proposed which recognises important aspects of the challenge of incorporating odour into LCA. While acknowledging the existence of barriers to complete odour assessment in LCA, we develop a midpoint indicator that goes beyond current approaches. We also provide new characterisation factors to operationalise the method and a model for calculation of additional factors for odourants that are not considered here.

1.1 A framework for odour assessment in LCA

Environmental impacts occur on “objects” or “areas” we wish to protect: human health, ecosystem functions and natural resources (Goedkoop et al. 2009). Traditionally, LCIA distinguishes between midpoint and endpoint indicators (Bare et al. 2000). Midpoint indicators are based on some aggregating procedure for the burden placed on the environment by different emissions without describing the actual extent of the environmental damage. The most well-known midpoint indicator is the carbon footprint, measured in the mass of carbon dioxide equivalents (Pandey et al. 2011). It describes the influence of a gas on planetary radiative forcing, but not the environmental consequences of this radiative forcing. On the other hand, it is feasible to characterise greenhouse gas emissions in terms of endpoint indicators that describe the potential impact on something people care more about. Examples include disability adjusted life years, a measure of human health impacts (de Schryver et al. 2009), potentially disappeared fraction of species per square metre and year (for lost ecosystems—see again de Schryver et al. 2009) or monetary, mass or energy units (for resources—see Klingemair et al. 2014).

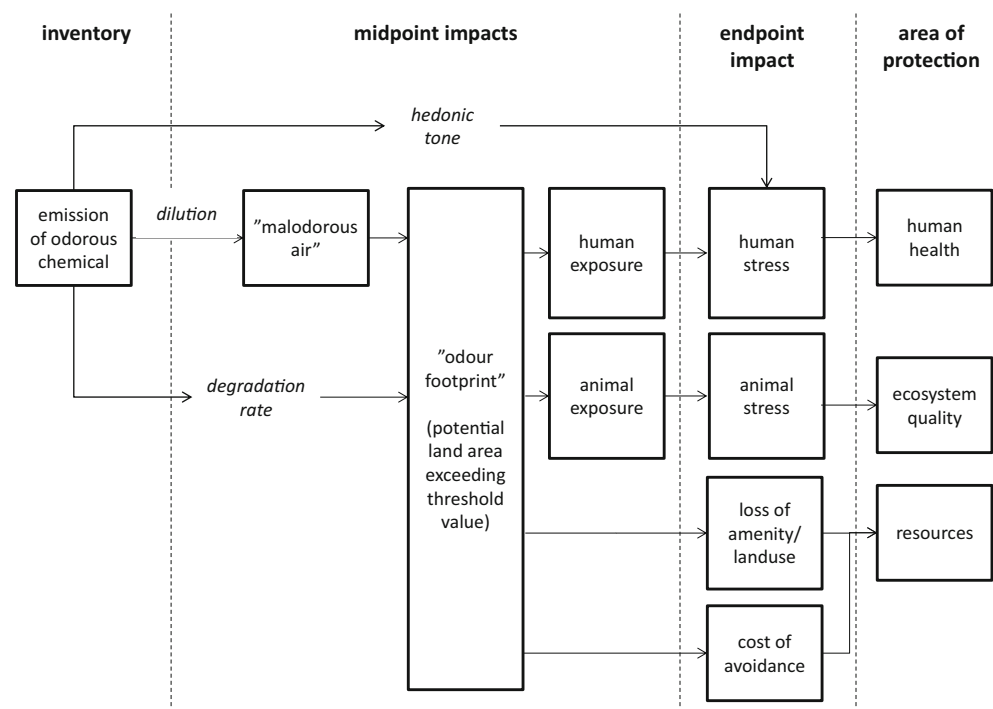
In the development of any LCIA approach for odour, a key question is with which effects and protection object(s)

Table 1 Summary of recent LCA studies where odour is taken into consideration

Article	Odour source	Key odourants	Assessment method	Notes
Kadam (2002)	Bagasse processing	Not listed	Volume odorous air (Guinee 2002)	Applied midpoint approach
Steinberg et al. (2004)	Biological waste treatment plant	Menthol, limonene, ethyl caprylate, ethylbutyrate, dimethylsulfide, alpha-pinene, acetophenone	Guinee 2002	Data for reduction of 7 odourants
Steinberg et al. (2005)	Biological waste treatment plant	2-Butanone, ethanol, limonene, alpha-pinene, dimethyl disulphide, ethyl butyrate, dimethyl sulphide	Characterisation factors used to rank odourants; case study then uses total organic carbon values as a proxy for odour	Authors compiled but did not publish 153 OTVs from Guinee (2002) and 2 reports in German
Jullien et al. (2006)	Asphalt pavements	Chrysene, benzo(a)fluoranthrene	Odour units per area and time	Odour flux as midpoint—individual chemicals not identified
Benetto et al. (2009)	Wood processing plant	Not listed	Odour units per volume for midpoint assessment	For the background LCIA methods Benetto refers to a website (Uni of Stuttgart, 2008) which is now lost.
Marchand et al. (2013)	Composting facility	Ethyl benzene	Odour units per mass relative to benzene for sweet odours	Suggested fate modelling using modified box model.

odorous emissions should be associated. As Fig. 1 illustrates, ideally, the impact of odorous emissions on any of the three traditional areas of protection would be considered if impacts occur. Leaving aside problems of data availability, if human health can be affected by odours, presumably, animal health can be too and these two protection objects should be considered. However, the relationship between perceptions of odour and effects on human health is difficult to characterise. This is

not to deny the fact that at higher doses, odourants can cause physical symptoms like eye irritation, headache, respiratory problems and nausea (Nicell 2009); but from an LCIA perspective, these toxicological effects of odorous chemicals are the domain of existing human toxicity potential indicators (see Goedkoop et al. 2009) and are not considered in Fig. 1. Humans can detect odours at parts per billion or trillion, orders of magnitude lower than those associated with toxicological

Fig. 1 A general framework for odour LCA

effects in mammals (Rosenkranz and Cunningham 2003). Nicell (2009) refers to such “sub-irritant levels” as causing “non-toxicological” effects. It may be feasible to propose a link between odour and health mediated by “non-toxicological” stress, since stress is capable of inducing its own pathologies. However, the potential to create an endpoint indicator based on odour causing human and animal “stress” in Fig. 1, is confounded by the highly variable human responses to stress, and stress adaptation in animals and people. Schiffman et al. (1995) describe a link between odour exposure and mood in a survey of self-reported symptoms including higher tension, depression, anger, fatigue and confusion among residents exposed to intermittent feedlot odours. They also point out connections between mood and the immune system. On the other hand, in a later controlled experiment, Schiffman et al. (2005) found neither measurable medical effects nor mood changes in participants exposed to odour from a pig house for 1 h, although exposed subjects did self-report headaches and nausea at concentrations below known effect thresholds. The researchers hypothesised that the lack of measurable effects was caused by participants’ knowledge that their exposure was brief and voluntary, while the self-reported effects were the consequence of a synergistic combination of irritants. Other researchers have claimed there is “no significant association between odour perceptibility and potential for inducing health effects” (Rosenkranz and Cunningham 2003, p15).

Given the fundamental challenges associated with connecting stress and exposure (shown in the upper right side of Fig. 1), future research might reasonably be directed towards the area of protection of resources. Resources can be assessed in financial terms in LCA—for example Steen (1999) used willingness-to-pay as a basis for weighting land use midpoint indicators. Estimating a cost associated with odour should be feasible in principle since odour impacts have financial consequences. One way to assess them is in terms of specific investments that are made to abate odours. Gathering data on such investments could provide a basis for an indicator for resource impacts (based on the idea that willingness-to-pay for odour abatement technology provides a proxy indicator of the value of the amenity provided by the land resource). This would be expected to depend at least indirectly on factors including the number of stakeholders downwind from the odour source and the difference in value of the affected land before and after the intervention (see “cost of avoidance” in Fig. 1). Another way would be to base such a metric directly on the change in land values associated with the loss of residential development in buffer zones created around odour sources like sewage treatment plants or poultry farms, an approach sometimes called “hedonic pricing”. A challenge to using this approach for endpoint indicator calculation is the need to find a robust average emission rate associated with the land price differential.

Considering the final step in the framework of Cucurachi et al. (2014), it appears that whether or not endpoint LCIA models considering impacts on resources are feasible, developing a midpoint indicator is a worthwhile step in this direction. Considering Fig. 1 in its entirety, it is apparent that all three areas of protection have an implicit or explicit dependence on an estimate of the potential land area above the odour detection threshold (“odour footprint”) which is associated with the loss of amenity or cost of avoidance. Reflecting the origin of the footprint concept in the ecological footprint (Pandey et al. 2011), the units of this indicator are areal. Such a midpoint should be more generally applicable than an endpoint indicator because it avoids being dependent on local and regional land values, which are highly location-specific. Furthermore, it would be more meaningful than Heijungs’ “malodorous air” which does not consider the persistence of odour. The midpoint indicator “odour footprint” is therefore the focus of the remainder of this article.

1.2 Models for odour

Gaussian plume models are routinely applied to odour sources such as sewage treatment facilities in order to calculate the statistical frequency of odour threshold exceedence in the surrounding landscape (e.g. Latos et al. 2011). Such models are based on the following equation:

$$C = \frac{Q}{u} \cdot \frac{f}{\sigma_y \sqrt{2\pi}} \cdot \frac{(g_1 + g_2 + g_3)}{\sigma_z \sqrt{2\pi}} \quad (2)$$

where C is the concentration of a given odourant at a given location, Q is the emission rate from a point source, u is the airspeed, f is the crosswind dispersion parameter, g_i are parameters taking into account vertical dispersion and reflection from temperature inversions and the ground, and σ_i are standard deviations of the emission distribution vertically ($i=z$) and horizontally ($i=y$) which are dependent on the distance from the source. Unfortunately, f and g_i are dependent on u , which is dependent on time and location, so it would be difficult to use this approach for generating a midpoint indicator without making arbitrary and/or time-and-location-specific climatic assumptions.

It should be remembered that LCIA midpoint indicators are not estimates of impacts, but estimates of emission burdens. Their intent is to compare the potential for different emitted pollutants to cause a problem and hence compare different systems which emit pollutants in a simplified way. This is why the current odour LCIA approach (“malodorous air”) is based on the idea that an emission of a mass of an odourant should be assessed as the volume of air it would occupy when diluted to its OTV. This is an application of the critical volumes’ approach to toxicants which has more in common with box modelling than with Gaussian plume models in use for detailed

local assessments by regulatory authorities licensing odour emissions.

It is feasible to add a temporal component to this box approach. USEtox is an example of a consensus LCIA approach based on multiple nested environmental compartments (or “boxes”) (Huijbregts et al. 2010). Using mixing height and reaction rate assumptions consistent with the USEtox model, one could calculate the (area \times time) impacted by a pulse of odourant. This would, of course, ignore the detailed mixing effects of wind and topography which must be considered in statutory environmental impact assessment, but would permit a ranking of contaminants on a more sophisticated way in LCA than is currently available. The scale of the initial atmospheric compartment in which the emission is diluted in such models is typically so large (USEtox version 1.01 uses $5.76 \times 10^{10} \text{ m}^3$ for emissions to urban air—see USEtox 2014) in comparison to the effect scales of odour emissions that any odour impact would probably be undetectable at the initial concentration, so a new “local” dilution would have to be introduced into such models to make this approach work—Marchand et al. (2013) suggested a box $50 \times 50 \text{ m}$ wide and 240 m high.

As an alternative to box modelling, one could model the time-dependent Brownian diffusion (Huerta-Cuellar et al. 2014) and spreading of such a pulse emission and consider the simultaneous effect of degradation in the atmosphere. Integrating the area and time over which an emission exceeds its OTV provides a proxy for the potential land area exceeding the OTV, and a relative scale on which to compare potential odourant impacts. Explicitly excluding wind speed and other site-specific factors allows the approach to focus on the chemical-specific factors controlling the extent of an emitted odourant’s potential impacts on people: diffusion, reaction and the OTV. Dilution by wind-induced turbulence obviously plays a key role in determining the actual effect of odourants outdoors, but it would have the same diluting effect on all odourants emitted from a single source, irrespective of their atmospheric lifetimes and human sensitivities. Thus, odour footprint (like “malodorous air”) can be understood as a non-site-specific midpoint indicator.

2 Method for odour footprinting

In the absence of wind or turbulent airflow, two factors can lead to the reduction of an odourant to a concentration below its OTV in both indoor and outdoor environments. These are: dilution by Brownian diffusion and chemical reaction. Although certain gases may react with other anthropogenic and natural gaseous reagents, the most important route for the destruction of organic odourants is reaction with hydroxyl radicals (Kwok and Atkinson 1995).

For the purpose of estimating an effected area, an initially homogeneous release of 1 kg of an odourant at the earth’s

surface in a hemisphere of initial radius 1 m was assumed. This geometry means the initial conditions were $V_{10}=2.09 \text{ m}^3$, $A_{10}=\pi \text{ m}^2$ and $C_{10}=477 \text{ g m}^{-3}$. A is the area within which the odourant exceeds its OTV. As shown in Fig. 2, the hemisphere will expand as the odourant diffuses and dilutes (path a along axis r in the figure). The increase is counteracted by simultaneous degradation reaction with hydroxyl radicals. The area exceeding the OTV reaches a maximum radius (the solid outline) and then collapses back to the origin (path b) due to the continued degradation process.

According to Fogler (1992), simultaneous diffusion and first-order reaction in a sphere can be represented by the following equation:

$$\frac{dC}{dt} = D \frac{\partial^2 C}{dr^2} + \frac{2D}{r} \frac{dC}{dr} - kC \quad (3)$$

where in this case, C is the concentration of the odourant, t is time, D is the Brownian diffusion coefficient obtained using Fuller’s method, r is the radial distance from the centre and k is the reaction coefficient. A finite difference approximation for the above equation was incorporated in a MATLAB™ program, where:

$$\frac{\partial^2 C}{dr^2} \cong \frac{C_{i+1} - 2C_i + C_{i-1}}{\Delta r^2} \quad (4)$$

$$\frac{dC}{dr} \cong \frac{C_i - C_{i-1}}{\Delta r} \quad (5)$$

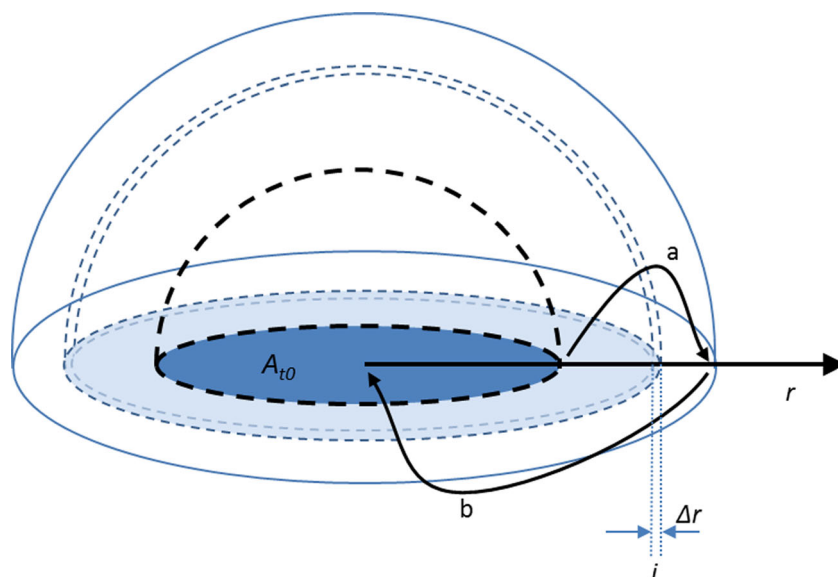
and i represents the position from the origin of a hemispherical segment of the hemisphere. This allows the calculation of a two-dimensional matrix $C_{t,r} [\text{g m}^{-3}]$ of the concentration of odourant in the atmosphere at finite timesteps Δt and finite distance Δr , analogous to fate factor matrix in classical multimedia modelling (denoted \overline{FF} in Rosenbaum et al. 2007) but with the environmental compartments as concentric segments (“ i ”) of a hemisphere. Since the intent of the model is to calculate the land area and time over which an odourant is capable of being detected by the human nose, the effect factor $EF_x = \text{OTV}_x^{-1} [\text{m}^3 \text{ g}^{-1}]$ for odourant x is multiplied with $C_{t,r}$ to scale this concentration matrix:

$$CF^*_{t,r} = C_{t,r} \text{OTV}^{-1} \quad (6)$$

Next, we identify the column vector $R^*_t = i$ for $CF^*_{t,r} = 1$ which is the segment at the limit of the odourant’s detectability by the human nose. From this, we can aggregate the area over time that exceeds this limit:

$$CF = \pi \sum_{t=0}^m (\Delta r R^*_t)^2 \quad (7)$$

Fig. 2 Expanding footprint concept



Practical values of m were obtained by trial and error, noting when C_{tr} fell to zero. We obtained a list of experimental values of k_{OH} from the NIST database (Manion et al. 2008). We used Atkinson's revised method for estimating k_{OH} based on structural contributions (Kwok and Atkinson 1995) to fill data gaps in this list (diethylamine and dimethyldisulfide). These data were all based on atmospheric pressure and 25 °C. Values of k were then calculated considering that a typical daytime value of the atmospheric concentration of hydroxyl radicals $[OH]$ is 1.5×10^6 molecules cm^{-3} (Allen 2001). Diffusion was assumed to only occur in the positive direction of r . A further boundary condition was assumed, that at a large value of r there was no concentration gradient ($C_x = C_{x+1}$). The model was stepped forward in increments of 1 cm and 1 s until all of the odourant had fallen below the OTV for all r and t . OTV values were taken from Abraham et al. (2012) who reproduced 193 values produced using Nagata's triangle odour bag method (Nagata 2003). We consider this to be a good source of this data compared to other available compilations of olfactometry results because the OTVs are corrected for recovery of single compounds and losses in the dynamic olfactometer are avoided (Hansen et al. 2013), and in contrast to other compilations, the Nagata data all are obtained in the same lab using the same method.

3 Results

Sample input values of D , k_{OH} and OTV are shown in Table 2 along with the results in terms of (area \times time) and the equivalent mass of hydrogen sulphide emitted. This gas was chosen as a benchmark because of its commonality and the relatively high public awareness of its status as an odourant.

Results for 33 additional odourants are presented in the [Supporting Information](#).

Some intermediate results produced during the model runs are shown in Fig. 3—the height of the curve is proportional to the square of the distance from the odourant source at which the odourant is above its OTV. The greater height of the curve (representing area exceeding OTV) for dimethyl sulphide (DMS) is caused by it having the lowest reaction rate of the four gases shown and the highest diffusion rate. Although the reaction rate for n-butanol is only 12 % higher than DMS, its OTV is 15 times higher; so it is not as detectable by the human nose. Methanethiol and p-cresol have relatively high and similar reaction rates (methanethiol is only 25 % lower than p-cresol). The factor of 2.5 between the sizes of their odour footprints is caused by methanethiol having a much higher diffusion rate and much lower OTV. Methanethiol has a much lower OTV than n-butanol, which, per se, would make it detectable further from the source; but its reaction rate is also much higher, giving a smaller area under the curve despite similar peak heights.

4 Case study

To illustrate the operation of this LCIA method, an LCA case study in odour abatement is provided.

4.1 Goal and scope

The question being examined here is whether installation of odour filtration equipment at a pig barn results in a net environmental benefit, or a mere transfer of environmental problems. The study is based on the difference between the odour-related impact of a pig house in Denmark with and

Table 2 Selected odourant parameters and results

	D	k_{OH}	OTV	Odour footprint	
	$m^2 s$	s^{-1}	$g m^3$	$m^2 day$	$kg H_2S\text{-eq } kg^{-1}$
Hydrogen sulphide	2.31E-05	7.05E-06	5.72E-07	1.60E+06	1.00E+00
Methanethiol	1.75E-05	4.97E-05	1.38E-07	5.87E+04	3.67E-02
p-cresol	1.07E-05	0.000066	2.43E-07	2.30E+04	1.44E-02
Dimethyl sulphide	1.46E-05	9.82E-06	7.62E-06	3.91E+05	2.45E-01
n-butanol	1.25E-05	1.1E-05	0.000115	1.56E+05	9.77E-02

without odour abatement technology supplied by the company SKOV®. In the base case, there is no treatment—exhaust fans merely expel the ventilated air containing odourants from the house. The technology of the biofilter case is described in detail in Liu et al. (2012). Briefly, it consists of a three-stage cellulose filter covered with biofilms wetted by a recirculating water irrigation system. The functional unit is the treatment of exhaust gas from a barn with four batches of 350 growing–finishing pigs over 1 year.

4.2 Life cycle inventory

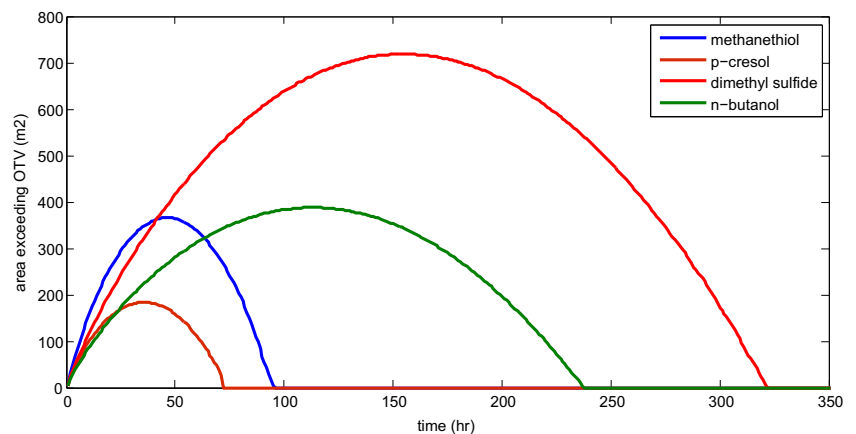
The LCI step leading to the odour footprint is no different to the usual LCI step in LCA, with the exception that the analyst must have some knowledge of the typical odourants for the product or process under assessment, and look for data on these. Which gases are relevant will depend on whether the study concerns wastewater treatment, high-density animal husbandry operations or other activities. For example, in this study, 24 odourants were considered relevant, and are listed in the Supporting Information (Table SI2) along with the results of the inventory modelling. Data on odour purification was taken directly from Liu et al. (2012): odourant emission rates before and after treatment, the pressure drop over the biofilter (40 Pa) and the filter irrigation rate (8.25 m³/h). The incremental energy consumption for ventilation and the biofilter was estimated at 13,160 kWh/year. For the base case, the

incremental energy consumption was estimated at 7,700 kWh/year. It was assumed that the irrigant purge rate in the biofilter case was optimised to prevent filter blockage, and therefore, that all the ammonia removed by the filter was distributed to farmers' fields, reducing the need for artificial fertilisation (ammonium nitrate) but causing ammonia emission in the fields equivalent to that caused by the avoided fertiliser. The filter material was assumed to be usable for 3 years. The cellulose was modelled as wood-free paper from an integrated mill and the electricity as Danish average power supply. Both of these background inventory elements were taken from the EcoInvent database, and the model was constructed in the Gabi 6.0 LCA software (PE International 2014) and presented in terms of kilogrammes of odourant emitted in Table SI2.

4.3 Life cycle impact assessment and interpretation

The impact categories chosen for this work are odour footprint, climate change potential and marine eutrophication potential, using ReCiPe 1.08 for the two last mentioned. These categories were chosen to illustrate the scale of the treatment effect and the impact of energy consumption and redirected nitrogen flows associated with the treatment intervention. The odourant inventory data were multiplied by the corresponding odour footprint characterisation factor in units of kilogrammes H₂S-e/kilogramme (shown in the last column

Fig. 3 Diffusive expansion and reactive decay of area exceeding OTV when 1 kg of selected gases is released



in Table S11 in the Supporting Information). The results of the life cycle assessment are shown in Table 3. The filter reduced the odour footprint by a factor of five. Odorous emissions at the farm were three orders of magnitude larger than emissions associated with power generation and production of filter material. Therefore, total odorous emissions were significantly lower in the biofilter case and burden shifting between locations is not important. The marine eutrophication potential in the biofilter case was a factor of almost five lower than in the base case. This impact category was mainly affected by ammonia emissions to the atmosphere and is therefore in line with the reduction of ammonia due to the biofilter. On the other hand, greenhouse emissions increased by approximately 2.5 tonnes of CO₂ equivalent per year. Electricity use was much higher when using both ventilation and a biofilter instead of ventilation only. Avoided fertiliser production in the biofilter case, where ammonia is captured and used as fertiliser, reduced the contribution to climate change, resulting in a net impact of 7.5 tonnes of CO₂ equivalent per year.

The key gas influencing the results was ammonia. This gas causes the vast majority of the odour impacts (98 % of the impact in the biofilter case, with the next most significant gas, hydrogen sulphide, accounting for 1 %). This was also true for the eutrophication potential; ammonia is responsible for 17.3 kg N-e/year (oxides of nitrogen were the next most significant, with 0.3 kg N-e/year in the biofilter case). Although the ammonia removed from the exhaust gas by the biofilter was applied on land, this was assumed to replace an equivalent amount of artificial fertiliser, resulting in no additional eutrophication. Likewise, although according to Ten Hoeve et al. (2014) about 12 % of the ammonia applied to the field would be expected to return to the gas phase (and cause further impacts) these emissions replaced emissions that the artificial fertiliser would have caused.

5 Discussion

In this article, we have tried to show a pathway for improvement of LCA methodology for odorous emissions. A

midpoint indicator was developed that expresses both the detectability and the persistence of an odorous emission. The article is meant as a step in the direction of a new impact category for odour, hopefully, one of many steps. While not intended as a complete assessment of an odourant removal technology, the case study demonstrated the kind of analysis that is feasible with this method, and showed a tradeoff between odourant impacts and greenhouse impacts.

More generally, the case study suggests that consideration of an odour indicator in LCA of odour control equipment is necessary. Odour reduction technologies are often installed with the sole purpose of reducing odour. An LCA ignoring the environmental impact of odour will always be unfavourable for odour reduction technologies as extra energy and materials are required. However, the case also demonstrated some shortcomings of the method developed. One weakness of this proposal, and the established methods for assessment of odour and toxic chemicals in LCA, is that the synergetic effects and impacts of degradation products are not considered. Also, although reaction with hydroxyl radicals is considered, the principal route for destruction of odourants in the atmosphere, other reactions may also play a role. For example, hydrogen sulphide reacts with oxygen at a rate of $2.4 \times 10^{-21} \text{ s}^{-1}$, much slower than the reaction with hydroxyl radicals described here ($7.1 \times 10^{-6} \text{ s}^{-1}$). Future work should consider other reactants and extend the list of characterisation factors to further odourants.

We consider that each of the prerequisites for consideration of the development of an LCIA method for a “novel” impact category suggested by Cucurachi et al. (2014) are satisfied by odourants. The key question is what the optimal approach to the LCIA modelling is. Impact assessment of odourant emissions is hampered by the local nature of the impacts. Emissions occurring in an area with few humans and animals have other consequences compared with emissions in densely populated areas, a fact reflected by LCIA methods with characterisation factors that differentiate between urban and rural settings (e.g. Lundie et al. 2007). A number of efforts to differentiate among local and regional environments using geography-based atmospheric models (e.g. Potting et al. 1998) or fluvial archetypes (e.g. Kounina et al. 2013) have also been made. This “localization” is a well-known challenge in LCA, but is especially relevant for odour. As suggested in the Introduction, one could develop our proposed odour footprint further by considering the value of the land affected by the odour. On the other hand, a balance has to be struck in LCA between making the method responsive to detail and efficient to apply. In this respect, our proposal is almost as easy for the analyst to apply as that of Heijungs et al. (1992). In the case of that method, the analyst need only divide LCI results by the appropriate characterisation factor (OTV—really an effect factor), and look in the literature for additional OTVs if any were missing from Heijungs’ list. In our method,

Table 3 Life cycle indicator results

	Odour footprint kg H ₂ S-e/year	Carbon footprint kg CO ₂ -e/year	Eutrophication kg N-e/year
Base case	11 110	4 818	84
Biofilter case (total)	2 056	7 514	18
Energy supply	14	12 176	1
Filter material	0.38	119	0
Farm emissions	2 080	0	18
Avoided fertiliser	−39	−4 781	−1

the analyst can also efficiently apply the characterisation factors listed in the [Supporting Information](#), while further factors can be calculated using a relatively simple model needing only D , k_{OH} and OTV as input data. On the other hand, greater complexity is apparent in the proposal of Marchand et al. (2013) which uses significantly more detail about the site at which emissions occur. That proposal involves running the USEtox model after changing up to 17 parameters in it for each emission site. This would be time consuming if the LCA analyst wishes to model an international supply chain with multiple emission locations. Another key innovation in Marchand et al. (2013) was the idea of disaggregating the characterised results into 11 different categories, each for a different type of smell (sweet, rancid, faecal etc.). That method may be appropriate in some circumstances but it could necessitate aggregation in a multicriteria analytical scheme (Rowley et al, 2012), so there will be a need for a midpoint method that is easier to implement and communicate, such as is described in this article.

6 Conclusions

Two principal criticisms of the current state of odour assessment in LCA can be made. One is that the OTVs listed in the principal guiding documents for LCA practitioners have not been updated with more accurate and consistent values since the early 90s. Another is that little work has been made to begin bringing chemical fate considerations into odour LCIA. This article takes steps towards improving LCIA for odours on both counts. It must be borne in mind that, in common with most LCIA methods, the odour footprint method proposed here does not calculate an actual impact, but is a way of comparing pollutant emissions with each other on a relative scale. It is a midpoint method which may be appropriate for use when odour impacts are considered significant in a system to be analysed using LCA, but a site-generic approach is needed.

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